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Nitrate impacts on groundwater from irrigated-vegetable systems in a humid north-central US sand plain

George J. Kraft*, Will Stites

College of Natural Resources, University of Wisconsin-Stevens Point, 1900 Franklin St., Stevens Point, WI 54481, USA

Abstract

Groundwater is frequently susceptible to nitrate pollution in irrigated regions possessing a humid climate, coarse soil, and shallow water table. Such pollution degrades local drinking water resources and increases watershed nitrate export. The irrigated-vegetable production agroecosystem of the Wisconsin Central Sand Plain (WCSP), north-central USA, exemplifies the problem. This study's research goal was to assess groundwater nitrate loading in this agroecosystem, with a view to manage groundwater quality and nitrate export in the WCSP and similar regions.

Nitrate loading was measured beneath a 44 ha field for 4 years using a novel "water-year" method, during three crops of sweet corn (*Zea mays* L.) and one of potato (*Solanum tuberosum* L.). Nitrate-N concentrations averaged 20 mg l^{-1} in shallow (upper 3 m) groundwater beneath the study field. Nitrate-N loading from the sweet corn ranged from 126 to 169 kg ha^{-1} per year, loading from potato was 228 kg ha^{-1} per year. Groundwater loading amounted to 61% of total available N and 77% of fertilizer N. Measured loadings compared well with those calculated using a budget approach, supporting the validity of both methods.

N budgets were calculated from average regional harvests and two fertilizer rates: a typical grower rate, and the more conservative University Extension rate. Budget-derived nitrate-N loadings from sweet corn are 151 kg ha^{-1} per year (typical) and 119 kg ha^{-1} per year (University Extension). For potato, typical and recommended fertilizer rates are equal, and the budget-derived nitrate-N loading is 203 kg ha^{-1} per year. Budget-derived loadings imply that limiting the basin-averaged nitrate-N concentration in groundwater to 10 mg l^{-1} (the US drinking water standard) would require each irrigated-vegetable hectare to be offset by 4.5–6.5 ha of land supplying nitrate-free groundwater recharge. Further, a WCSP watershed with half irrigated-vegetable cover, even with no other nitrate sources, would export 50–74 kg ha^{-1} per year of nitrate-N. This export is greater on a per-hectare basis than the large maize- and soybean-producing basins of Iowa, USA.

Most orthodox nitrate control strategies (e.g., decreasing and splitting fertilizer, irrigation scheduling) have already been implemented in the WCSP, and have little further potential to substantially reduce loading if profitability must be maximized. Better control will require new and unconventional approaches for controlling nitrate leaching from mineralized crop residue as well as additional curbing of direct fertilizer nitrogen leaching.
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1. Introduction

1.1. Agricultural impacts on groundwater

* Corresponding author. Tel.: +1-715-346-2984;
fax: +1-715-346-2965.
E-mail address: gkraft@uwsp.edu (G.J. Kraft).

Agricultural groundwater pollution from nitrate is a worldwide problem that has economic, ecosystem,

and human health impacts (O'Neil and Raucher, 1990; USEPA, 1990; Spalding and Exner, 1993; Goolsby, 2000). Besides degrading drinking water resources (Hamilton and Helsel, 1995), aquatic ecosystems are affected when pollutant-bearing groundwater discharges to surface water. Nitrate can harm the eggs and young of some salmonids and amphibians (Kincheloe et al., 1979; Hecnar, 1995; Johnston et al., 1999; Marco et al., 1999; Rouse et al., 1999), promote eutrophication in N-limited freshwaters, and increase growth of rooted aquatic plants (Lillie and Barko, 1990; Rodgers et al., 1995). Nitrate export from freshwater basins can cause eutrophication in saltwater bodies. The groundwater discharging from the study area, for instance, eventually reaches the Mississippi River and contributes to hypoxia problems in the Gulf of Mexico (e.g., Goolsby, 2000).

Sandy irrigated agricultural areas in humid regions such as the north-central USA are particularly susceptible to groundwater nitrate pollution (Mossbarger and Yost, 1989). Irrigation has expanded eightfold in the north-central USA since about 1970 (Bajwa et al., 1992), mainly in sandy areas. This expansion has been accompanied by substantial nitrate pollution (Mossbarger and Yost, 1989). For instance, in portions of a Wisconsin Central Sand Plain (WCSP) (Fig. 1) groundwater basin affected by irrigated-vegetable fields, nitrate-N averaged $13.7 \, \text{mg} \, \text{l}^{-1}$ compared with $0.5 \, \text{mg} \, \text{l}^{-1}$ for unaffected groundwater (Kraft et al., 1999). Nitrate-N exports from one WCSP basin are presently $22 \, \text{kg} \, \text{ha}^{-1}$ (unpublished data) and increasing, among the highest in the central USA (Goolsby, 2000).

A correlation between agriculture and nitrate groundwater pollution is well established (Hamilton and Helsel, 1995; Kolpin, 1997). What is frequently lacking for managing such pollution are the tools to measure and predict the nitrate loading to groundwater from particular agricultural systems. Such tools would permit nitrate concentration and export trends to be predicted, strategy building for meeting groundwater quality goals, an understanding of local contributions to continent-scale nitrate loading to marine environments, and assessment of the carrying capacity of a physical setting for certain agricultural systems.

This study's research goal was to assess groundwater nitrate loading from the irrigated-vegetable production system in the WCSP, north-central USA (Fig. 1),
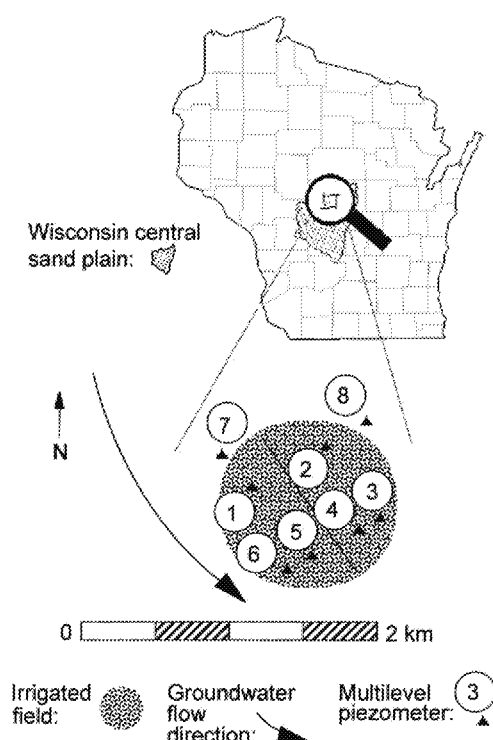

Fig. 1. Location of study area and field instrumentation.

with a view to manage nitrate groundwater quality and export in this and similar regions. This required developing tools for measuring and predicting nitrate loading for this setting. Specific study objectives included measuring nitrate loading to groundwater beneath a 44 ha field over a 4-year period using a novel "water-year" method, comparing measured loadings to those calculated using a budget approach, and applying the budget method to explore broader management concerns.

2. Study area

2.1. Geology, soils, climate, and groundwater conditions

The WCSP (Fig. 1) is a $6400 \, \text{km}^2$ area characterized by level topography and a mantle of coarse-grained Pleistocene sediment frequently more than 30 m thick overlying low permeability bedrock. Upland soils that

developed in the Pleistocene sediment are sandy and extremely well drained. Topsoils average 93% sand and 1% organic matter, while subsoils average 98% sand and 0.1% organic matter. The aquifer in the Pleistocene sediment supplies water for irrigation and industry, and both rural and municipal drinking water. Aquifer materials typically contain less than 0.1% organic matter and 92–95% quartz. Groundwater is well oxygenated except for the deepest parts of the aquifer at a few locations and adjacent to some wetlands (Kraft et al., 1999).

The climate of the WCSP is humid, temperate, and continental (Bartelme, 1977; Otter and Fiala, 1978). Winters are cold and snowy; summers are generally warm. The average frost-free growing season is 133 days (Bartelme, 1977). Annual precipitation averages about 790 mm. Approximately 60% of the annual precipitation falls in May–September. Of 790 mm precipitation, 510–560 mm goes to evapotranspiration, 230–255 mm recharges groundwater, and 25 mm runs off (Weeks et al., 1965; Holt, 1965).

2.2. Irrigated-vegetable production system

The irrigated-vegetable rotation in the area consists of 1 year of potato and 2–3 years of sweet corn, snap bean (*Phaseolus vulgaris* L.), field corn (*Zea mays* L.), soybean (*Glycine max* L.), or pea (*Pisum sativum* L.), in that order of frequency (D. Sexson, 1999, pers. commun.). Typical fertilizer-N applications to these crops are (respectively) 258, 200, 110, 180, 67, and 67 kg ha^{-1}. Nitrogen fertilizer is typically applied as ammonium nitrate, urea, or ammonium sulfate; and K fertilizer as potassium chloride. Manure is rarely ap-

plied. This production system is managed at a level close to the current state of the art with respect to irrigation scheduling and timing of fertilizer applications, though not necessarily fertilizer amounts.

Planting usually occurs between mid-April (for early potato) and early June. Fertilizer is applied at planting and 2–6 more times by banding and with irrigation water. Irrigation is by center-pivot sprinkler. Depending on weather, irrigation water is applied about 15 times per year, about 15 mm per irrigation. Irrigation increases the transpiration of fields by about 100 mm per year, and also increases percolation and solute leaching (Weeks et al., 1965; Weeks and Stangland, 1971). Nonadsorbing solutes applied to fields usually leach to the water table during the year of application (e.g., Kung, 1990).

3. Study field, crops, cultural practices, and weather in 1992–1995

3.1. Study field, crops, and cultural practices

The study field (Fig. 1) is 44 ha in the WCSP. The soil is Plainfield loamy sand (mixed, mesic Typic Udipsamment). Underlying Pleistocene deposits consist of about 20 m of well-sorted medium sand with a 1 m interbedded fine-textured layer about 7.5 m below the surface. Groundwater occurs at 2.5–3 m depth, and flows southeast at about 0.1 m per day.

Sweet corn was grown in 1992, 1994, and 1995, and potato was grown in 1993. Agricultural management decisions including nutrient application rates and timing (Table 1 and Fig. 2) were made by the

Table 1
Crops, fertilizer-N inputs, and harvest amounts during the 4-year study period

Year	Crop	N fertilizer input (kg ha^{-1})			Harvest (Mg ha^{-1}, fresh)	
		Actual	Recommended[a]	Typical[b]	Actual	Typical[c]
1992	Sweet corn	250	168	200	25	20
1993	Potato	297/357[d]	258	258	46	41.5
1994	Sweet corn	207	168	200	21	20
1995	Sweet corn	176	168	200	20	20

[a] University Extension-recommended input.
[b] Typical grower input.
[c] Typical harvest for region.
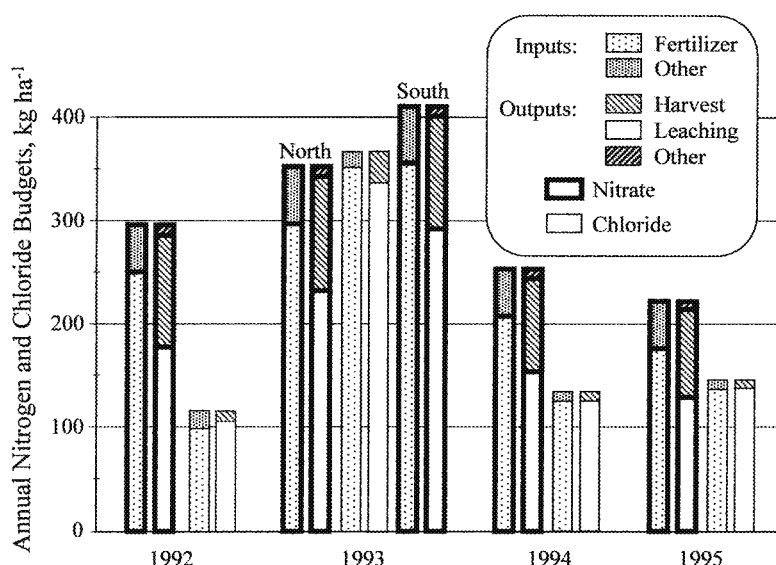[d] North/south field halves.

Fig. 2. Nitrogen and chloride budgets for study field.

cooperating commercial operator. Sweet corn crops were planted in mid to late May and harvested 84–95 days later. Harvests were average to above average for the region (Table 1). The 1993 potato crop was planted on 1 May and harvested 151 days later. Harvest was above average. Annual fertilizer applications started with potassium chloride fertilizer applied 36–76 days before planting. NPK plus micronutrient fertilizer was applied at planting, and additional N was applied five or six times during the growing season. Unusually heavy rains in 1993 resulted in wet conditions in the north field half and prevented the grower from applying as much fertilizer in the north half as in the south half (297 kg ha^{-1} vs. 357 kg ha^{-1}).

3.2. Weather

The 1992 growing season was shorter (97 days) and cooler (106 degree-days, 18 °C basis) than average (133 days and 234 degree-days; Bartelme, 1977) with average precipitation. The 1993 growing season had about average length (128 days) and heat units (198 degree-days) but was unusually wet. Rainfall in May and June was 129 mm more than average. The 1994 growing season was about average in precipi-

tation and heat units but somewhat short (117 days). The 1995 growing season was about average in length and precipitation, but warm (373 degree-days).

4. Methods

4.1. Field instrumentation

Six multilevel piezometers (MLPs) were installed in the field to monitor groundwater (Fig. 1). Two more MLPs were installed 20 m upgradient of the field to obtain groundwater unaffected by agriculture. Upgradient land uses were nonagricultural, and provided groundwater recharge low in nitrate, and usually low in chloride. In-field MLPs were placed more than a year's groundwater travel distance from upgradient field edges, so that a complete year of recharge from the field would be available within the sampled zone.

MLPs had 19 screens 15–40 cm long over a continuous length of 340 cm. The heavy rains in 1993 caused the water table to rise above the zone sampled by the MLPs. To accommodate the higher water table, supplementary MLPs were added next to existing ones. These contained up to five screens at 25 cm intervals.

4.2. Sampling and analysis

Groundwater was sampled for nitrate and chloride 38 times at 2-week to 2-month intervals from January 1992–March 1996. Usually only alternate ports of each MLP were sampled. MLP tubes were purged (three times static volume) and sampled with a peristaltic pump. Samples were filtered (0.45 μm) and preserved with 0.25% (v/v) concentrated sulfuric acid, and kept on ice until delivered to the laboratory. Nitrate and chloride were analyzed by automated colorimetry. The analytical method for nitrate includes nitrite, but nitrite is undetectable or minute in this environment (R. Stephens, University of Wisconsin-Stevens Point, unpublished data). Hence, the results are reported simply as nitrate-N.

4.3. Statistics

When a single concentration value was needed to characterize each MLP, weighted averages based on port lengths were used. When sampled ports were not contiguous, the interval between ports was divided evenly between the adjacent ports. Time averages were weighted according to intervals between sampling dates. Comparisons between years used 1 May–30 April period, corresponding roughly to crop years and first appearance of spring-applied fertilizer chloride at the water table.

Nonparametric statistics (Kendall rank correlation statistic, Kruskal–Wallis test, and Wilcoxon signed-rank test) were used because most parameters were not normally distributed. The type I error level for significance was 0.05.

4.4. Nitrate and chloride loading

4.4.1. Water-year method

The water-year method (Stites and Kraft, 2001) estimates the loading of nonadsorbing solutes through frequent and detailed monitoring of a vertical groundwater profile. This was accomplished in this study using the MLPs. The water-year was about the same as the crop year.

The theory behind the water-year method is illustrated in Fig. 3. All groundwater recharge during a year in a region of a field with area A (large enough to average out heterogeneities) will occupy a prism P. The top and bottom surfaces of P also have area A. The prism's vertical axis is tilted because deeper, older recharge water has moved farther downgradient than the shallower, younger water. The volume of water in P is $\theta A(z_2 - z_1)$, where θ is the volumetric water content of the aquifer, and z_2 and z_1 the elevations of the upper and lower faces of P, respectively. If a solute's concentration C were uniform with depth, the mass m of solute in P would be $\theta AC(z_2 - z_1)$; rearranging and integrating in the vertical direction to account for the non-uniform concentration yields

$$\frac{m}{A} = \theta \int_{z_1}^{z_2} C \, \mathrm{d}z \qquad (1)$$

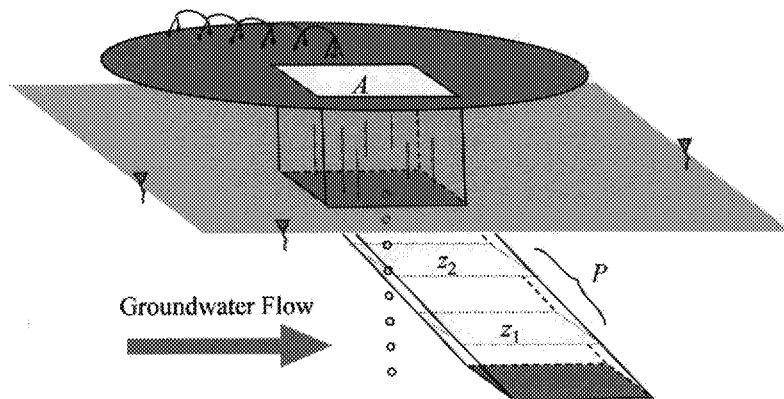where m/A is the annual mass loading per unit area.

Fig. 3. Conceptual model of water-year method. A: unit area of field; P: prism containing 1 year's groundwater recharge originating in A; z_1 and z_2 are the elevations of the lower and upper surfaces of P, respectively.
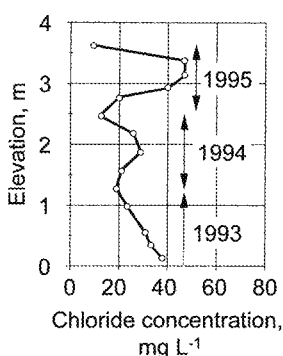
Fig. 4. Chloride concentration profiles in MLP 6 on 29 August 1995, showing downward movement of pulses. Elevations are relative to the bottom of the sampled zone. Arrows indicate bands of annual recharge.

The challenge to implementing the water-year method is to discern z_1 and z_2—the depths in the aquifer that correspond to the beginning and end of a water-year. Chloride proved to be a satisfactory tracer for this purpose. It was applied to the field at the start of each water-year as potassium chloride fertilizer, 36–76 days before planting, and leached to the water table within months. The arrival of a year's chloride pulse at the water table and its subsequent downward movement in the aquifer were readily apparent in a time series of concentration–depth profiles for each MLP. Fig. 4 illustrates that all or part of three water-years were present in the monitored zone on 29 August 1995. The entire 1995 Cl pulse is fully within the sampled zone, between 2.5 and 3.6 m elevation. These elevations correspond to z_1 and z_2 for the 1995 water-year in Eq. (1). The 1994 Cl pulse is between 1.3 and 2.5 m, and the 1993 pulse has begun to exit the sampled zone. Later profiles showed continued downward movement of Cl pulses.

Operationally, the water-year method was implemented by plotting chloride concentration–depth profiles for each sampling event and then arranging them as a time series for each MLP. When chloride minima in the monitored zone bracketed a year's water, the mass of a solute of interest (nitrate and Cl) for the water-year could be established. Loading estimates were computed using Eq. (2), a discrete version of Eq. (1):

$$\frac{m}{A} = \theta \sum_{p=i}^{j} C_p \Delta z_p \qquad (2)$$

where i and j are the bottom and top MLP ports in the range from z_1 to z_2, C_p the concentration in port p, and Δz_p the length of port p. (For ports at the boundary between water-years, half of the port's length was assigned to each year.) Aquifer volumetric water content (θ) was 0.40 (Kraft et al., 1995).

Usually a given water-year interval remained in the monitored zone during several sampling events before it was displaced by younger water. Hence several loading estimates could be made at a single MLP. Groundwater was flowing horizontally past MLPs between sampling events, and later sampling events encountered groundwater that had originated farther upgradient than earlier sampling events. Thus, water-year profiles sampled on different dates represented slightly different locations in the field, helping to represent in-field variability.

4.4.2. Nitrate budget

The budget method of Meisinger and Randall (1991) calculates long-term potentially leachable total nitrogen, N_{pl} as

$$N_{pl} = N_{input} - N_{output} - \Delta N_{st} \qquad (3)$$

where N_{input} and N_{output} are N entering and leaving the field between the top of the crop canopy and the bottom of the root zone, respectively, and ΔN_{st} the change in N storage. N_{pl} was used as the budget-derived estimate of nitrate-N loading to groundwater during a crop year. This is justified because nitrate and other nonadsorbing solutes leach rapidly (within months) to groundwater in this setting.

Nitrogen inputs considered were fertilizer, crop seed, and atmospheric deposition. Sweet corn seed supplied a negligible $0.3 \, kg \, N \, ha^{-1}$, and potato "seed" $9 \, kg \, N \, ha^{-1}$ (Meisinger and Randall, 1991). Atmospheric deposition was estimated at $20 \, kg \, ha^{-1}$ per year, based on a review of work by Hoeft et al. (1972) and others.

The largest N output was crop harvest. Harvested N was calculated from crop yield and N concentration. For sweet corn, a concentration of $4.3 \, g \, kg^{-1}$ (fresh weight) was used, based on Meisinger and Randall (1991). Local data were available for potato (Bundy et al., 1997; Saffigna et al., 1977; Wilner et al., 1997), with a fresh-weight concentration range of $2.0–2.8 \, g \, kg^{-1}$ and a mean of 2.4.

Fertilizer volatilization and denitrification outputs were judged negligible based on previous studies (Saffigna et al., 1977; Oberle and Bundy, 1987), as was erosion (Bartelme, 1977). Miscellaneous gaseous N losses from soil and crop were estimated to be $7 \, \text{kg} \, \text{ha}^{-1}$ for sweet corn and $9 \, \text{kg} \, \text{ha}^{-1}$ for potato (Meisinger and Randall, 1991).

ΔN_{st} has three potential components: inorganic N, crop residue, and soil organic matter. Inorganic and crop-residue N were assumed to be the same at the beginning and end of the budget year. Therefore, only soil organic matter mineralization contributes to ΔN_{st}. Oberle and Keeney's (1990) estimated soil organic N mineralization on Plainfield loamy sand, corrected for estimated atmospheric N deposition, indicates $\Delta N_{st} = -25 \, \text{kg} \, \text{ha}^{-1}$ per year.

4.4.3. Chloride budget

A similar budget for chloride was calculated and compared to water-year chloride loadings as an additional check on the water-year method. Chloride inputs considered were fertilizer (potassium chloride), atmospheric deposition, and irrigation water. Outputs were crop harvest and loading to groundwater. Year-to-year change in chloride storage was assumed to be zero. Annual chloride inputs from fertilizer in 1992–1995 were 99, 353, 125, and $137 \, \text{kg} \, \text{ha}^{-1}$. Reported atmospheric chloride inputs in the area range from 1 to $7 \, \text{kg} \, \text{ha}^{-1}$ per year (Cabrera-Rivera, 1989; MacDonald et al., 1992; Quideau and Bockheim, 1997). Irrigation chloride input was estimated using chloride concentrations in nearby monitoring wells. Crop harvest is the only significant chloride output other than leaching. Potato chloride concentrations inferred from Saffigna et al. (1977) imply exports of $22–37 \, \text{kg} \, \text{ha}^{-1}$. No chloride concentration data were found for sweet corn. Limited information on field corn (Parker et al., 1985) suggests harvest export between 15 and $45 \, \text{kg} \, \text{ha}^{-1}$.

5. Results

5.1. Nitrate and chloride concentrations

During the 4 years of monitoring, concentrations in individual groundwater samples beneath the field ranged from <0.2 to $50.5 \, \text{mg} \, \text{l}^{-1}$ nitrate-N and $<1–119 \, \text{mg} \, \text{l}^{-1}$ chloride. The field's monitored zone averaged about $20 \, \text{mg} \, \text{l}^{-1}$ of both nitrate-N and chloride over the period. Concentrations in the monitored zone varied among MLPs and over time, ranging from 9.1 to $32.8 \, \text{mg} \, \text{l}^{-1}$ nitrate-N, and $4.4–46.5 \, \text{mg} \, \text{l}^{-1}$ chloride. Upgradient MLPs had much smaller concentrations of nitrate and chloride than in-field MLPs, averaging $\leq 0.8 \, \text{mg} \, \text{l}^{-1}$ nitrate-N and $\leq 1.8 \, \text{mg} \, \text{l}^{-1}$ chloride over the 4 years.

5.2. Nitrate and chloride loading

5.2.1. Water-year method

Annual water-year nitrate-N loadings averaged $172 \, \text{kg} \, \text{ha}^{-1}$ per year: $126–169 \, \text{kg} \, \text{ha}^{-1}$ for the three sweet corn crops and $228 \, \text{kg} \, \text{ha}^{-1}$ for potato. Annual chloride loadings averaged $182 \, \text{kg} \, \text{ha}^{-1}$ per year with a range of 111–366 (Fig. 2).

Budget and water-year loading estimates agreed well (medium crop-concentration estimate, Table 2). The two methods were within 5% for the cumulative

Table 2
Nitrate-N and chloride loading to groundwater beneath an irrigated-vegetable field in central Wisconsin

Year	Loading ($\text{kg} \, \text{ha}^{-1}$)					
	Nitrate-N			Chloride		
	Water-year		Budget	Water-year		Budget
	Mean ± S.E.	Range[a]		Mean ± S.E.	Range[a]	
1992	165 ± 11	137–207	179	111 ± 11	87–156	106
1993	228 ± 12	187–261	263[b]	366 ± 20	322–448	337
1994	169 ± 24	122–285	154	116 ± 07	99–140	125
1995	126 ± 10	91–153	129	135 ± 11	108–186	138

[a] Range of values at individual MLPs, each of which is averaged over several sampling dates.
[b] Average of north and south field-halves.

4-year nitrate loading, with a maximum difference of 15% in any year. Chloride matched within 3% over the entire period and 8% in any one year.

6. Discussion

6.1. Nitrate concentrations

Previous studies in the WCSP have shown greater nitrate concentrations in shallow groundwater beneath irrigated-vegetable fields than was observed in this study. Nitrate concentrations were roughly 25–100% greater beneath three fields studied by Curwen et al. (1991) (unpublished report) and two fields studied by Nehls et al. (2001). Perhaps concentrations were smaller beneath the study field because incoming groundwater underflow contained little nitrate. As a result, little nitrate was cycled through the field in irrigation water. In addition, a large vertical concentration gradient between groundwater originating in the field and incoming underflow would promote dispersion of nitrate deeper into the aquifer.

6.2. Nitrate and chloride loading

Variability of loading estimates among MLPs (Table 2) was about the same as variability among sampling dates at individual MLPs. Individual MLPs were not consistently above or below the field average. Likely causes of variability are uneven chemical applications, redistribution after application, or uneven (e.g., focused) recharge (Derby and Knighton, 2001).

Nitrate loading (from water-year values; budget results are similar) amounted to 61% of the total N input and 77% of fertilizer input during the study. Chloride loading (water-year method) amounted to 95% of total input and 102% of fertilizer input. The potato crop supplied more nitrate to groundwater, but about the same proportions of fertilizer N and total N (Fig. 2). Nitrate loading among the sweet corn crops increased with increasing fertilizer. An inverse relationship between sweet corn yield and nitrate loading was expected (because larger harvests should remove more N from the field) but no such relationship was observed. Yield effects were probably masked by the great variation in fertilizer inputs among years.

The budget-derived and water-year loading estimates agreed well for both nitrate and chloride. This indicates that both methods are useful for estimating loading in this setting. The largest difference between the two methods was in 1993. Denitrification may explain the difference that year, when copious early summer rainfall kept the soil waterlogged for weeks. Denitrification was excluded from the budget because it is normally negligible in this setting (Saffigna and Keeney, 1977), but during that exceptionally wet year it may have removed a measurable amount of nitrate-N.

6.3. Applying budget method to typical inputs and yields

The budget method can be extended to estimate nitrate loading with selected yields and fertilizer rates. For sweet corn, the typical fertilizer-N input in this setting is about $200\,kg\,ha^{-1}$ (W. Ebert, USDA Natural Resources Conservation Service, unpublished producer survey), whereas the University Extension recommends N $168\,kg\,ha^{-1}$ (Binning et al., 2000). Based on a long-term average yield of $20\,Mg\,ha^{-1}$ at experimental farms (L.G. Bundy, pers. commun., 1999), the predicted nitrate-N loading is $151\,kg\,ha^{-1}$ (typical input) or $119\,kg\,ha^{-1}$ (recommended input). For potato, the typical fertilizer-N input (equal to the University Extension recommendation) is $258\,kg\,ha^{-1}$ (Exo, 1993). Based on a potato harvest of $41.5\,Mg\,ha^{-1}$ (1992–1996 average; USDA NASS, 1998), the nitrate-N loading is $203\,kg\,ha^{-1}$ for the typical (or recommended) input.

6.4. Obstacles to reducing nitrate loading

Three major strategies have been used to reduce nitrate leaching in this agricultural system. *Decreasing total fertilizer nitrogen* eliminates part of the soil nitrogen that is likely to be lost to leaching. *Splitting fertilizer nitrogen inputs* reduces the size of the soil nitrate pool available for leaching at any one time. *Irrigation management* aims to prevent leaching from excessive irrigation water being added to antecedent soil water. All these strategies are implemented within the constraint of maximizing crop profitability. These strategies do not sufficiently control nitrate loading to groundwater in this setting. Better control of nitrate

loading will require additional control of direct fertilizer nitrogen leaching as well as control of nitrate leaching from mineralized crop residue. ("Direct" leaching refers to nitrate from fertilizer that leaches without being taken up by a crop.)

Direct fertilizer N leaching is substantial, especially for potato. Potato needs a sustained high soil nitrate level for yield and quality (Sattelmacher and Marschner, 1979) and it has a shallow root system. WCSP potato plots lost 53% of fertilizer N to direct leaching with a University Extension-recommended N application of 224 kg ha^{-1} (Andraski and Bundy, 1999). Sweet corn may be less susceptible to direct leaching, but the losses are still substantial. Andraski and Bundy (1999) measured 34% direct leaching of 190 kg ha^{-1} fertilizer N (approximately the typical amount). Direct leaching is difficult to control, partly because soils in the region have small water holding capacities. Irrigation exacerbates the water holding capacity problem, because it is used to keep soil moisture near the optimum for plant growth. Therefore a heavy rainfall can quickly leach out most of the nitrate in the root zone.

Nitrate loss from mineralized crop residue has received little attention. Sweet corn residue contains about half the aboveground N taken up (e.g., 84 kg N ha^{-1}; Andraski and Bundy, 1999). Estimates of potato residue N in this region range widely, from 10 to 99 kg ha^{-1} (Saffigna and Keeney, 1977; Bundy et al., 1997), with 40 kg ha^{-1} perhaps typical. Crop residues quickly mineralize to nitrate in this setting (Andraski and Bundy, 1999), and the mineralized nitrate apparently leaches below the root zone before a subsequent crop can capture it. Andraski and Bundy (1999) found almost no residue N (1–2%) was taken up by the next season's crop, i.e., almost all residue N ultimately becomes available for leaching.

In contrast to fertilizer nitrogen recoveries of 31–52% in harvested tubers found in this and other studies (Andraski and Bundy, 1999) in this setting, very high recoveries have been reported in the western USA, ca. 80–90% (Tyler et al., 1983; Westermann et al., 1988). Such high recoveries apparently depend on little rainfall during the growing season, as well as minimal leaching from crop residue.

Options to reduce nitrate loading are limited. Nitrification inhibitors might reduce direct leaching, but they decrease potato yield and quality (Kelling, 1998). Reducing fertilizer-N inputs enough to appreciably affect nitrate loading would reduce crop yields, making it unlikely that growers could remain competitive unless they can command a premium price for crops grown with reduced chemical inputs. Removing residues from fields is also uneconomic, unless new uses for the residues are developed. Furthermore, removing crop residues aggravates wind erosion and degrades soil quality. Changing to a crop rotation that is less dependent on N fertilizer could be beneficial, but again maintaining profitability is problematic. Coexistence of high-quality groundwater with irrigated-vegetable production in this setting appears to require either a fundamental technological advance or a willingness by society to subsidize more sustainable production methods.

6.5. Regional implications

Estimates of nitrate loading can be extended to assess nitrate effects on the scale of a basin or region. Assuming University Extension fertilizer-N recommendations and typical harvests, a potato–sweet corn–sweet corn rotation would load 147 kg ha^{-1} per year of nitrate-N to groundwater. Different third year crops would probably load less nitrate to groundwater than sweet corn (Mechenich and Kraft, 1996). However, even if the third year crop loaded no nitrate, the 3-year rotation would still load greater than 100 kg N ha^{-1} per year. Nitrate is generally conserved in WCSP groundwater (i.e., denitrification is not important), so dilution is the only mechanism available for maintaining basin-average groundwater quality. To limit nitrate-N to 10 mg l^{-1}, the US Maximum Contaminant Level (MCL, a drinking water standard), the amount of land required to offset irrigated-vegetable land can be calculated from the following equation:

$$10 \text{ mg l}^{-1} \text{ nitrate-N} = \frac{L_i A_i}{R_i A_i + R_{ni} A_{ni}} \qquad (4)$$

where L is the N loading rate (M L^{-2} T^{-1}), R the net recharge rate (L T^{-1}), A the land area (L^2) and subscripts i and ni refer to irrigated and nonirrigated, respectively. This assumes recharge from nonirrigated land contains no nitrate. Solving for A_{ni} indicates that each hectare of irrigated-vegetable land that loads 100–147 kg ha^{-1} per year of nitrate-N to groundwater

needs to be offset with 4.5–6.5 ha of land that loads no nitrate.

Nitrate export from agricultural basins affects distant ecosystems. A WCSP groundwater basin where half the land cover is irrigated-vegetable agriculture would export 50–74 kg ha^{-1} per year nitrate-N from irrigated fields alone. Goolsby et al. (2001) stated that the agricultural basins contributing the most to eutrophic hypoxia in the Gulf of Mexico supplied 15–31 kg ha^{-1} per year of nitrate-N. It appears that basins in the study area may be creating a larger downstream impact per hectare than the large basins in the US maize and soybean belt.

7. Conclusions

Nitrate-N concentrations averaged about 20 mg l^{-1} in the shallow (upper 3 m) groundwater beneath the study field. Expected concentrations under WCSP fields are typically greater. Nitrate-N loading to groundwater averaged 172 kg ha^{-1} per year over 4 years. The 3 years of sweet corn loaded 126–169 kg ha^{-1} while the year of potato loaded 228 kg ha^{-1}. Nitrate-N loaded to groundwater amounted to 61% of total available N and 77% of fertilizer N. Compared with sweet corn, potato loaded more N to groundwater, but about the same proportions of fertilizer N and available N.

The expected nitrate-N loading in this agroecosystem from sweet corn with average yield and typical fertilizer rates is 151, or 119 kg ha^{-1} with University Extension-recommended rates. For potato, the typical and recommended fertilizer-N rates are about the same, and would result in an expected nitrate-N load of 203 kg ha^{-1}. The expected nitrate-N loading to groundwater from a potato–sweet corn–sweet corn rotation is 147 kg ha^{-1} per year, using Extension-recommended N fertilizer rates. Even with a third year crop or fallow conditions that would load no nitrate to groundwater, the 3-year rotation would still load greater than 100 kg ha^{-1} per year. Given these loadings and that nitrate is generally conservative in WSCP groundwater, dilution by low-nitrate groundwater recharge is the only mechanism available for maintaining basin-average nitrate concentration below some management standard. Limiting basin-average nitrate-N concentration to the US MCL (10 mg l^{-1}) would require offsetting each irrigated-vegetable hectare with 4.5–6.5 ha of land supplying nitrate-free recharge.

The WCSP vegetable production system apparently produces a large export flux of nitrate to surface water. A typical WCSP basin with half its land cover in irrigated-vegetable production would export 50–74 kg nitrate-N ha^{-1} per year. Hence, WCSP watersheds may create a larger downstream impact per hectare than the large maize- and soybean-producing basins of Iowa, USA.

Options to control nitrate loading are limited. Orthodox strategies such as decreasing fertilization rates, splitting fertilizer inputs, and optimizing irrigation scheduling have little further potential to reduce loading if profitability must be maximized. Better control will require controlling nitrate leaching from mineralized crop residue as well as additional curbing of direct fertilizer nitrogen leaching.

References

Andraski, T.W., Bundy, L.G., 1999. Nitrogen cycling in crop residues and cover crops on an irrigated sandy soil. Agron. Abstr., p. 244.

Bajwa, R.S., Crosswhite, W.M., Hostetler, J.E., Wright, O.W., 1992. Agricultural irrigation and water use. Agriculture Information Bulletin No. 638. US Department of Agriculture, Economic Research Service, Rockville, MD.

Bartelme, R.J., 1977. Soil Survey of Wood County, Wisconsin. US Department of Agriculture, Soil Conservation Service, Washington, DC.

Binning, L.K., Boerboom, C.M., Bundy, L.G., Delahaut, K.A., Harrison, H.C., Kelling, K.A., Mahr, D.L., Mahr, S.E.R., Michaelis, B.A., Stevenson, W.R., Wedberg, J.L., Wyman, J.A., 2000. Commercial vegetable production in Wisconsin-2000. University of Wisconsin-Extension Bulletin A3422. Cooperative Extension Publishing, Madison, WI.

Bundy, L.G., Andraski, T.W., Bland, W.L., 1997. Nitrogen cycling in crop residues and cover crops. Proc. Wis. Ann. Potato Mtgs. 10, 53–61.

Cabrera-Rivera, O., 1989. Chemical characteristics of wet deposition in Wisconsin, 1980–1986. Wisconsin DNR PUBL-AM-031-89, 51 pp.

Curwen, D., Kraft, G., Osborne, T., Shaw, B., 1991. Demonstration of low input strategies for potato/vegetable production on irrigated sands. Final Report. Wis. Dep. Agric., Trade, Consumer Protection Sust. Agric. Demonstration Proj. #8810.

Derby, N.E., Knighton, R.E., 2001. Field-scale preferential transport of water and chloride tracer by depression-focused recharge. J. Environ. Qual. 30, 194–199.

Exo, J., 1993. Farm Practices Inventory. Unpub. Survey. University of Wisconsin-Extension, Madison, WI.

Goolsby, D.A., 2000. Mississippi basin nitrogen flux believed to cause Gulf hypoxia: EOS. Trans. Am. Geophys. Union 81, 321–327.

Goolsby, D.A., Battaglin, W.A., Aulenbach, B.T., Hooper, R.P., 2001. Nitrogen input to the Gulf of Mexico. J. Environ. Qual. 30, 329–336.

Hamilton, P.A., Helsel, D.A., 1995. Effects of agriculture on ground-water quality in five regions of the United States. Ground Water 33, 217–226.

Hecnar, S.J., 1995. Acute and chronic toxicity of ammonium nitrate fertilizer to amphibians from southern Ontario. Environ. Toxicol. Chem. 14, 2131–2137.

Hoeft, R.G., Keeney, D.R., Walsh, L.M., 1972. Nitrogen and sulfur in precipitation and sulfur dioxide in the atmosphere in Wisconsin. J. Environ. Qual. 1, 203–208.

Holt Jr., C.L.R., 1965. Geology and Water Resources of Portage County Wisconsin. US Geological Survey Water-supply Paper 1796. US Government Printing Office, Washington, DC.

Johnston, N.T., Stamford, M.D., Ashley, K.I., Tsumura, K., 1999. Responses of rainbow trout (*Oncorhynchus mykiss*) and their prey to inorganic fertilization of an oligotrophic montane lake. Can. J. Fish. Aquat. Sci. 56, 1011–1025.

Kelling, K.A., 1998. Using nitrification inhibitors for increasing N use efficiency on potatoes. Proc. Wis. Ann. Potato Mtgs. 11, 93–95.

Kincheloe, J.W., Wedemeyer, G.A., Koch, D.L., 1979. Tolerance of developing salmonid eggs and fry to nitrate exposure. Bull. Environ. Contam. Toxicol. 23, 575–578.

Kolpin, D.W., 1997. Agricultural chemicals in groundwater of the midwestern United States: relations to land use. J. Environ. Qual. 26, 1025–1037.

Kraft, G.J., Stites, W., Mechenich, D.J., Balma, J., 1995. Port Edwards Groundwater Priority Watershed, Groundwater Resource and Agricultural Practice Evaluation. Central Wisconsin Groundwater Center, University of Wisconsin-Stevens Point.

Kraft, G.J., Stites, W., Mechenich, D.J., 1999. Impacts of irrigated vegetable agriculture on a humid north-central US sand plain aquifer. Ground Water 37, 572–580.

Kung, K.-J.S., 1990. Preferential flow in a sandy vadose zone. 1. Field observation. Geoderma 46, 51–58.

Lillie, R.A., Barko, J.W., 1990. Influence of sediment and groundwater on the distribution and biomass of *Myriophyllum spicatum* L. in Devil's Lake, Wisconsin. J. Freshwater Ecol. 5, 417–426.

MacDonald, N.W., Burton, A.J., Liechty, H.O., Witter, J.A., Pregitzer, K.S., Mroz, G.D., Richter, D.D., 1992. Ion leaching in forest ecosystems along a Great Lakes air pollution gradient. J. Environ. Qual. 21, 614–623.

Marco, A., Quilchano, C., Blaustein, A.R., 1999. Sensitivity to nitrate and nitrite in pond-breeding amphibians from the Pacific northwest, USA. Environ. Toxicol. Chem. 18, 2836–2839.

Mechenich, D.J., Kraft, G.J., 1996. Contaminant source assessment and management using groundwater flow and contaminant models in the Stevens Point-Whiting-Plover wellhead protection area. Unpublished Report. Central Wisconsin Groundwater Center, University of Wisconsin-Stevens Point/University of Wisconsin-Extension.

Meisinger, J.J., Randall, G.W., 1991. Estimating nitrogen budgets for soil–crop systems. In: Follett, R.F. (Ed.), Managing Nitrogen for Groundwater Quality and Farm Profitability. ASA, CSSA, SSSA, Madison, WI, pp. 85–124.

Mossbarger, W.A., Yost, R.W., 1989. Effects of irrigated agriculture on groundwater quality in Corn Belt and Lake States. J. Irrig. Drain. Eng. 115, 773–790.

Nehls, T., Arriaga, F., Kelling, K.A., Lowery, B., 2001. Nitrate loading under different N rates and surfactants and potato yield. In: Proceedings of the Wis. Ann. Potato Mtgs., vol. 14. University of Wisconsin-Extension, Madison, WI, pp. 79–85.

Oberle, S.L., Bundy, L.G., 1987. Ammonium volatilization from nitrogen fertilizers surface-applied to corn (*Zea mays*) and grass pasture (*Dactylis glomerata*). Biol. Fert. Soils 4, 185–192.

Oberle, S.L., Keeney, D.R., 1990. Soil type, precipitation, and fertilizer N effects on corn yields. J. Prod. Agric. 3, 522–527.

O'Neil, W.B., Raucher, R.S., 1990. The costs of groundwater contamination. J. Soil Water Conserv. 45, 180–183.

Otter, A.J., Fiala, W.D., 1978. Soil Survey of Portage County, Wisconsin. US Department of Agriculture, Soil Conservation Service, Washington, DC.

Parker, M.B., Gaines, T.P., Gascho, G.J., 1985. Chloride effects on corn. Commun. Soil Sci. Plant Anal. 16, 1319–1333.

Quideau, S.A., Bockheim, J.G., 1997. Biogeochemical cycling following planting to red pine on a sandy prairie soil. J. Environ. Qual. 26, 1167–1175.

Rodgers, S.J., McFarland, D.G., Barko, J.W., 1995. Evaluation of the Growth of *Vallisneria americana* Michx. Lake Reservoir Manage. 11 (1), 57–66.

Rouse, J.D., Bishop, C.A., Struger, J., 1999. Nitrogen pollution: an assessment of its threat to amphibian survival. Environ. Health Persp. 107, 799–803.

Saffigna, P.G., Keeney, D.R., 1977. Nitrogen and chloride uptake by irrigated Russet Burbank potatoes. Agron. J. 69, 258–264.

Saffigna, P.G., Keeney, D.R., Tanner, C.B., 1977. Nitrogen, chloride, and water balance with irrigated Russet Burbank potatoes in a sandy soil. Agron. J. 69, 251–257.

Sattelmacher, B., Marschner, H., 1979. Tuberization in potato plants as affected by applications of nitrogen to the roots and leaves. Potato Res. 22, 49–57.

Spalding, R.F., Exner, M.E., 1993. Occurrence of nitrate in groundwater: a review. J. Environ. Qual. 22, 392–402.

Stites, W., Kraft, G.J., 2001. Nitrate and chloride loading to groundwater from an irrigated north-central US sand-plain vegetable field. J. Environ. Qual. 30, 1176–1184.

Tyler, K.B., Broadbent, F.E., Bishop, J.C., 1983. Efficiency of nitrogen uptake by potatoes. Am. Potato J. 60, 261–269.

USDA NASS (National Agricultural Statistics Service), 1998. Crops county data (online). http://usda.mannlib.cornell.edu/data-sets/crops/9X100/ (verified January 3, 2002).

USEPA (United States Environmental Protection Agency), 1990. The drinking water criteria document on nitrate/nitrite. National Technical Information Service Document No. PB91-142836.

Weeks, E.P., Stangland, H.G., 1971. Effects of irrigation on streamflow in the central sand plain of Wisconsin. US Dept. of Interior Open-File Report.

Weeks, E.P., Ericson, D.W., Holt Jr., C.L.R., 1965. Hydrology of the Little Plover River basin Portage County, Wisconsin, and the effects of water resource development. US Geological Survey Water-Supply Paper 1811. US Government Printing Office, Washington, DC.

Westermann, D.T., Kleinkopf, G.E., Porter, L.K., 1988. Nitrogen fertilizer efficiencies on potatoes. Am. Potato J. 65, 377–386.

Wilner, S.A., Kelling, K.A., Massie, L.R., 1997. Influence of nitrogen timing and irrigation methods on Russet Burbank potatoes. Proc. Wis. Ann. Potato Mtgs. 10, 43–52.